Appendix E: Ecological Effects of Criteria Pollutants

Introduction

Benefits to human welfare from air pollution reductions achieved under the CAA can be expected to arise from likely improvements in the health of aquatic and terrestrial ecosystems and the myriad of ecological services they provide. For example, improvements in water quality stemming from a reduction in acid deposition-related air pollutants (e.g., SO_x and NO_x) could benefit human welfare through enhancements in certain consumptive services such as commercial and recreational fishing, as well as non-consumptive services such as wildlife viewing, maintenance of biodiversity, and nutrient cycling. Increased growth and productivity of U.S. forests could result from reduced emissions of ozone-forming precursors, particularly VOCs and NO_x, and thus may yield benefits from increased timber production; greater opportunities for recreational services such as hunting, camping, wildlife observation; and nonuse benefits such as nutrient cycling, temporary CO_2 sequestration, and existence value.

In this Appendix, the potential ecological benefits from CAA pollutant controls are discussed in the context of three types of ecosystems: aquatic, wetland, and forest. In describing the potential ecological benefits of the CAA, it is clearly recognized that this discussion is far from being comprehensive in terms of the types and magnitude of ecological benefits that may actually have occurred from the implementation of the CAA. Rather, this discussion reflects current limitations in understanding and quantifying the linkages which exist between air quality and ecological services, in addition to limitations in the subsequent valuation of these services in monetary terms. This discussion also does not cover potential benefits from improvements in other ecological services, namely agriculture and visibility, which are discussed and quantified in other sections of this report. This appendix is dedicated to a *qualitative* evaluation of ecological benefits. However, where possible, the existing body of scientific literature is drawn upon in an attempt to provide insights to the possible magnitude of benefits that may have resulted from CAA-related improvements of selected ecological services. It is important to note that the inability to fully value ecological services results in a significant undervaluation of the ecological benefits of air pollution reductions. This undervaluation should not be interpreted as a devaluation.

Benefits From Avoidance of Damages to Aquatic Ecosystems

Aquatic ecosystems (lakes, streams, rivers, estuaries, coastal areas) provide a diverse range of services that benefit the welfare of the human population. Commercially, aquatic ecosystems provide a valuable food source to humans (e.g., commercial fish and shellfish harvesting), are used for the transportation of goods and services, serve as important drinking water sources, and are used extensively for irrigation and industrial processes (e.g., cooling water, electrical generation). Recreationally, water bodies provide important services that include recreational fishing, boating, swimming, and wildlife viewing. They also provide numerous indirect services such as nutrient cycling, and the maintenance of biological diversity.

Clearly, these and other services of aquatic ecosystems would not be expected to be equally responsive to changes in air pollution resulting from the implementation of the CAA. The available scientific information suggests that the CAA-regulated pollutants that can be most clearly linked to effects on aquatic resources include SO_x and NO_x (through acid deposition and increases in trace element bioavailability), NO_x (through eutrophication of nitrogen-limited water bodies), and mercury (through changes in atmospheric deposition). Potential benefits from each of these processes (acid deposition, eutrophication, mercury accumulation in fish) are described separately in the following sections.

Acid Deposition

Background

Acid deposition refers to the depositing of strong acids (e.g., H_2SO_4, HNO_3) and weak acids ($(NH_4)_2SO_4$, NH_4NO_3) from the atmosphere to the earth's surface. Acid deposition can occur in the wet or dry form and can adversely affect aquatic resources through the acidification of water bodies and watersheds. Acidification of aquatic ecosystems is of primary concern because of the adverse effects of low pH and associated high aluminum concentrations on fish and other aquatic organisms. Low pH can produce direct effects on organisms, through physiological stress and toxicity processes, and indirect effects, mediated by population and community changes within aquatic ecosystems. Acidification can affect many different aquatic organisms and communities. As pH decreases to 5.5, species richness in the phytoplankton, zooplankton, and benthic invertebrate communities decreases.[1] Additional decreases in pH affect species richness more significantly, and may sometimes affect overall biomass.[2] Table E-1 presents descriptions of the biological effects of acidification at different pH levels. In evaluating the severity of biological changes due to acidification, the reversibility of any changes is an important consideration; biological populations and communities may not readily recover from improved water quality under certain circumstances. Researchers have addressed acidification effects through many different experimental protocols, including laboratory bioassays, particularly concerning pH, aluminum, and calcium; manipulative whole-system acidification studies in the field; and comparative, nonmanipulative field studies.

Although acidification affects phytoplankton, zooplankton, benthic invertebrates, fish, amphibians, and waterfowl, most acidification research has concentrated on fish populations.[3] Aluminum, which can be toxic to organisms, is soluble at low pH and is leached from watershed soils by acidic deposition.[4] Acidification may affect fish in several ways. The direct physiological effects of low pH and high aluminum include increased fish mortality, decreased growth, and decreased reproductive potential. The mechanism of toxicity involves impaired ion regulation at the gill.[5] Population losses occur frequently because of recruitment failure,[6] specifically due to increased mortality of early life stages.[7] Changes at other trophic levels may affect fish populations by altering food availability.[8] Fish in poorly buffered, low pH water bodies may accumulate higher levels of mercury, a toxic metal, than in less acidic water bodies, due to increased mercury bioavailability. The primary consequence of mercury accumulation appears to be hazardous levels to humans and wildlife who consume fish, rather than direct harm to aquatic organisms (discussed further below).

The CAA-regulated pollutants that are likely to have the greatest effect on aquatic ecosystems through acid deposition and acidification are SO_2 and NO_x. In the atmosphere, SO_2 and NO_x react to form sulfate and nitrate particulates, which may be dry-deposited; also the pollutants may react with water and be wet-deposited as dilute sulfuric and nitric acids. SO_2 is considered the primary cause of acidic deposition, contributing 75 to 95 percent of the acidity in rainfall in the eastern United States.[9]

Current Impacts of Acid Deposition

Effects on Water Chemistry

The effects of acid deposition and resulting acidification of water bodies was intensively studied as part of a 10-year, congressionally-mandated study of acid rain problems in the United States.[10] Based on the NAPAP study, it is estimated that 4 percent of the lakes and 8 percent of the streams in acid-sensitive

[1] J. Baker et al., NAPAP SOS/T 13, 1990; Locke, 1993.

[2] J. Baker et al., NAPAP SOS/T 13, 1990.

[3] NAPAP, 1991.

[4] J. Baker et al., NAPAP SOS/T 13, 1990.

[5] J. Baker et al., NAPAP SOS/T 13, 1990.

[6] Rosseland, 1986.

[7] J. Baker et al., NAPAP SOS/T 13, 1990.

[8] Mills et al., 1987.

[9] NAPAP, 1991.

[10] NAPAP, 1991.

Table E-1. Summary of Biological Changes with Surface Water Acidification.

pH Decrease	Biological Effects
6.5 to 6.0	Small decrease in species richness of phytoplankton, zooplankton, and benthic invertebrate communities resulting from the loss of some acid-sensitive species, but no measurable change in total community abundance or production. Some adverse effects (decreased reproductive success) may occur for acid-sensitive fish species (e.g., fathead minnow, striped bass).
6.0 to 5.5	Loss of sensitive species of minnows and dace, such as blacknose dace and fathead minnow; in some waters decreased reproductive success of lake trout and walleye. Distinct decrease in the species richness and change in species composition of the phytoplankton, zooplankton, and benthic invertebrate communities. Loss of a number of common invertebrate species from the zooplankton and benthic invertebrate communities, including zooplankton species such as *Diaptomus silicis*, *Mysis relicta*, *Epischura lacustris*; many species of snails, clams, mayflies, and amphipods, and some crayfish. Visual accumulations of filamentous green algae in the littoral zone of many lakes and in some streams.
5.5 to 5.0	Loss of several important sport fish species, including lake trout, walleye, rainbow trout, and smallmouth bass; as well as additional non-game species such as creek chub. Continued shift in the species composition and decline in species richness of the phytoplankton, periphyton, zooplankton, and benthic invertebrate communities; decreases in the total abundance and biomass of benthic invertebrates and zooplankton may occur in some waters. Loss of several additional invertebrate species common in oligotrophic waters, including *Daphnia galeata mendotae*, *Diaphanosoma leuchtenbergianum*, *Asplancha priodonta*; all snails, most species of clams, and many species of mayflies, stoneflies, and other benthic invertebrates. Inhibition of nitrification. Further increase in the extent and abundance of filamentous green algae in lake littoral areas and streams.
5.0 to 4.5	Loss of most fish species, including most important sport fish species such as brook trout and Atlantic salmon. Measurable decline in the whole-system rates of decomposition of some forms of organic matter, potentially resulting in decreased rates of nutrient cycling. Substantial decrease in the number of species of zooplankton and benthic invertebrates, including loss of all clams and many insects and crustaceans; measurable decrease in the total community biomass of zooplankton and benthic invertebrates in most waters. Further decline in the species richness of the phytoplankton and periphyton communities. Reproductive failure of some acid-sensitive species of amphibians such as spotted salamanders, Jefferson salamanders, and the leopard frog.

Source: Baker, J. et al. (NAPAP SOS/T 13, 1990), p. 13-173.

regions of the U.S. are chronically acidic due to natural and anthropogenic causes. NAPAP defines acidic conditions as occurring when the acid neutralizing capacity[11] (ANC) is below 0 µeq/L. Furthermore, approximately 20 percent of the streams and lakes in these regions are considered to be extremely susceptible to acidity (defined as ANC <50 µeq/L) and slightly more than half show some susceptibility to acidification (defined as ANC <200 µeq/L).

In terms of the role of acid deposition as a causal mechanism for the acidification of water bodies, it is estimated that 75 percent of the 1,181 acidic lakes and 47 percent of the 4,668 streams studied under

[11] ANC is expressed in units of microequivalents per liter (µeq/L), where an equivalent ANC is the capacity to neutralize one mole of H^+ ions. Generally, waters with an ANC < 0 have corresponding pH values of less than 5.5 (L. Baker et al., NAPAP SOS/T 9, 1990).

NAPAP receive their dominant source of acid anions from atmospheric deposition (see Table E-2). On a regional basis, the importance of acid deposition varies considerably, which is believed to result from regional differences in SO_x and NO_x emissions and differences in the biogeochemistry of individual watersheds. For acidic lakes (ANC <0), the regions that appear most likely to be influenced by acid deposition include the Adirondacks and Mid-Atlantic Highland region, with acid deposition cited as the dominant source of acidity in 100 percent of the acidic lakes studied (Table E-2). This is in stark contrast to the West region, where none of the acidic lakes studied were dominated by acid deposition (notably, the sample size of lakes for this region was small to begin with). For acidic streams, the Mid-Atlantic Highland region contains the greatest proportion of streams whose acidic inputs are dominated by acid deposition (56 percent). This contrasts with acidic streams of Florida, where the vast majority (79 percent) are acidic primarily due to organic acids, rather than acid deposition.

Table E-2. Comparison of Population of Acidic National Surface Water Survey (NSWS) by Chemical Category[1]

Region	Number of Acidic Waters	Deposition Dominated (%)	Organic Dominated (%)	Acid Mine Drainage Dominated (%)	Watershed Sulfate Dominated (%)
LAKES					
New England	173	79	21	--	--
Adirondacks	181	100	--	--	--
Mid-Atlantic Highlands	88	100	--	--	--
Southeastern Highlands	--	--	--	--	--
Florida	477	59	37	--	4
Upper Midwest	247	73	24	--	3
West	15	--	--	--	100
All Lakes	1,181	75	22	--	3
STREAMS					
Mid-Atlantic Highlands	2,414	56	--	44	--
Mid-Atlantic Coastal Plain	1,334	44	54	--	2
Southeastern Highlands	243	50	--	50	--
Florida	677	21	79	--	--
All Streams	4,668	47	27	26	<1

[1] Source: NAPAP 1991 (Table 2.2-3, p. 28).

Effects on Fish Habitat Quality

By combining information on relevant water chemistry parameters (pH, aluminum, calcium), fish toxicity models, and historical and current distributions of fish populations in the lakes and streams included in the National Surface Water Survey (NSWS), NAPAP investigators estimated the proportion of water bodies with water chemistry conditions that are unsuitable for survival of various fish species.[12] In the Adirondack region, where the acidic lakes are dominated by acid deposition, it is estimated that ten percent of the lakes are unsuitable for the survival of acid-tolerant fish species such as brook trout; twenty percent of the lakes are estimated to be unsuitable for

[12] NAPAP, 1991.

the survival of acid-sensitive species such as minnows. About two percent and six percent of the lakes in the New England region are estimated to be unsuitable for acid-tolerant and acid-sensitive fish species, respectively. A greater proportion of streams in the Mid-Atlantic Highland region are estimated to be unsuitable for acid-tolerant and acid-resistant fish species (18 percent and 30 percent, respectively); however, about 44 percent of streams surveyed in this region are thought to be heavily influenced by acid mine drainage (Table E-2).

Economic Damages to Recreational Fishing

In an effort to assess some of the impacts from *existing* levels of acid deposition to public welfare, NAPAP investigated the current economic damages associated with acid deposition to trout anglers of New York, Maine, Vermont, and New Hampshire. The general approach used consisted of linking the catch per unit effort (CPUE) for four species of trout at individual lakes (estimated using participation survey data) to the relevant water quality conditions at these lakes (namely, the acid stress index or ASI). Using historical water quality data, critical water quality conditions (i.e., the ASI values) were estimated for lakes in the absence of acid deposition and compared to current conditions reflecting the presence of acid deposition. Using two types of travel cost models, the Random Utility Model (RUM) and Hedonic travel-cost model (HTCM), estimates of the willingness to pay (WTP) per trip of sampled trout anglers were obtained. Aggregate estimates of the WTP were obtained across the populations of trout anglers using statistical weighting factors. Finally, the difference in total WTP between the current (acid deposition) scenario and the historical (acid deposition-free) scenarios was determined.

The resulting estimates of economic damages to trout anglers in the four state region are relatively small. Specifically, damage estimates range from $0.3 million to $1.8 million (in 1989 dollars) for the hedonic travel-cost and random utility models, respectively. By many accounts, these estimates can be considered to underestimate actual damages to anglers in these states. First, data limitations precluded the development of meaningful WTP estimates for brook trout anglers, which may be a significant component of trout fishing in these areas. Second, resource constraints necessitated exclusion of a large population of trout anglers (i.e., those residing in New York City). Third, the economic damage estimates were limited to trout anglers, thus excluding potentially similar if not greater economic damages to anglers fishing for other coldwater or warmwater fish species. In addition, the NAPAP analysis was performed in the context of recreational fishing in lakes, thereby excluding potentially important welfare impacts from recreational fishing in streams. Finally, these estimates do not address non-use values of lakes in this region.

Benefits From Acid Deposition Avoidance Under the CAA

It is currently estimated that in the absence of pollution reductions achieved under the Clean Air Act, total sulfur emissions to the atmosphere would have increased by nearly sixteen million tons by 1990, a 40 percent increase above 1990 levels estimated with CAA controls remaining in place.[13] Based on atmospheric transport and deposition modeling, this increase in sulfur emissions corresponds to an approximate 25 to 35 percent increase in total sulfur deposition (wet & dry) in large portions of the northeastern portion of the United States.[14] Given sulfur emission and deposition changes of this magnitude, and the importance of sulfur emissions in contributing to acid deposition, one would expect some benefits to human welfare to be achieved as a result of improved quality of aquatic ecosystems. To date, however, no formal benefits assessment of CAA-avoided acid deposition impacts has been conducted for aquatic ecosystems. Nevertheless, past benefit assessments involving acid deposition impacts on aquatic ecosystems provide some opportunity to gain insights into the relative magnitude of certain aquatic-based benefits that may be achieved through pollution reductions under the CAA.[15]

Recreational Fishing

NAPAP evaluated the impact of changes in acid deposition on use values of aquatic ecosystems (i.e., recreational fishing).[16] In their integrated assessment, NAPAP valued the impacts of three different sulfur-

[13] U.S. EPA, 1995; Table B-2.

[14] U.S. EPA 1995, p. 3-10.

[15] See, for example, NAPAP, 1991.

[16] NAPAP, 1991.

induced acid deposition scenarios to trout anglers from NY, VT, NH and ME.[17] The three scenarios evaluated were:

1. No change in acid deposition.
2. A 50 percent reduction in acid deposition.
3. A 30 percent increase in acid deposition.

As described above, equations were developed by NAPAP to estimate the catch per hour for species at each lake as a function of the ASI value for each lake and of the technique of the fishers. Baseline and predicted changes in CPUE were evaluated for all lakes modeled in the region. Willingness-to-pay estimates for CPUE per trip were derived for the baseline and sulfur emission scenarios using two travel-cost models, a random utility model and a hedonic travel cost model. These willingness-to-pay estimates were then combined with the results of a participation model that predicted the total number of trips taken by trout anglers. Total welfare changes were determined over a 50 year period (from 1990 to 2040).

At current levels of acid deposition, NAPAP estimates that trout anglers in these four states will experience annual losses by the year 2030 of $5.3 or $27.5 million (in 1989 dollars) for the random utility model and hedonic travel cost model, respectively (see Table E-3). If acid deposition *increases* by 30 percent, which roughly corresponds to the 25 to 35 percent increase predicted for the northeast U.S. in the absence of CAA sulfur controls,[18] the resulting economic losses to trout anglers in 2030 would range from $10 million to nearly $100 million annually (in 1989 dollars) for the RUM and HTCM, respectively. If deposition decreases by 50 percent, annual benefits to recreational anglers are estimated to be $14.7 million (RUM) or $4.2 million (HTCM).

While an estimation of CAA-related benefits to trout anglers based on the 30 percent increase in acid deposition scenario has some appeal, a strict transfer of these benefits to the section 812 retrospective analysis is hindered by several factors. First, the NAPAP benefits estimates are projected for future conditions (the year 2030). Therefore, the extent to which the NAPAP benefits reflect conditions and benefits in 1990 (the focus of the section 812 retrospective assessment) is unclear. Second, the NAPAP and CAA section 812 analyses operate from different baselines (1990 for the NAPAP study versus 1970-1990 for the section 812 study). However, the NAPAP estimates of annual benefits of $10 to $100 million provide a rough benchmark for assessing the likely magnitude of the avoided damages to an important and sensitive recreational fishery in a four-state area most impacted by surface water acidification from atmospheric deposition.

Table E-3. Results from Benefits Assessments of Aquatic Ecosystem Use Values from Acid Deposition Avoidance.

Study	Use Value	Scenario Modeled	Method	Annual Benefits
NAPAP (1991)	Trout Fishing	No change in acid deposition	RUM HTCM	-$5.3 million - $27.5 million
	(NY, ME, VT, NH)	50% decrease in acid deposition	RUM HTCM	$14.4 million $4.2 million
		30% increase in acid deposition	RUM HTCM	-$10.3 million -$97.7 million
		No new emission reductions after 1985	RUM HTCM	-$5.5 million -$3.5 million
		10 million ton reduction of SO_2 from 1980 levels by 2000	RUM HTCM	$9.7 million $4.4 million

Eutrophication

Eutrophication is the process by which aquatic systems respond to nutrient enrichment. The most common nutrients involved in eutrophication are nitrogen and phosphorous (and related chemical species). When water bodies receive excessive amounts of nutrients, adverse impacts on their resident species and on ecosystem functions can occur from excessive algal growth and the reduction in dissolved oxygen caused by decaying algal biomass. Under highly eutrophic conditions, excessive nutrients can cause depleted oxygen levels that result in subsequent loss of economically important benthic organisms (shellfish), fish kills, and changes in phytoplankton, zooplankton, and fish community

[17] NAPAP, 1991; p. 383-384.

[18] U.S. EPA, 1995.

composition.[19] Nuisance algal blooms can have numerous economic and biological costs, including water quality deterioration affecting biological resources, toxicity to vertebrates and higher invertebrates, and decreased recreational and aesthetic value of waters.[20] Although severe eutrophication is likely to adversely affect organisms, especially fish, a moderate increase in nutrient levels may also increase fish stocks, by increasing productivity in the food chain.[21]

Atmospheric Deposition and Eutrophication

The deposition of NO_x in aquatic systems and their watersheds is one source of nitrogen that may contribute to eutrophication. The relative importance of NO_x deposition as a contributor to aquatic eutrophication depends on the extent to which the productivity of an aquatic ecosystem is limited by nitrogen availability and the relative importance of nitrogen deposition compared to other internal and external sources of nitrogen to the aquatic ecosystem. Furthermore, the vulnerability of aquatic ecosystems to eutrophication is known to vary seasonally and spatially, although these systems are affected by nutrient deposition throughout the year. In general, freshwater ecosystems appear to be more often limited by phosphorus, rather than nitrogen, and are not as likely to be heavily impacted by nitrogen deposition compared to some estuarine and coastal ecosystems.[22] In contrast to acidification of streams and lakes, eutrophication from atmospheric deposition of nitrogen is more commonly found in coastal and estuarine ecosystems, which are more frequently nitrogen-limited.[23]

Unfortunately, there is limited information with regard to the relative importance of atmospheric deposition as a nitrogen source in many estuarine and marine ecosystems. Estimates of the importance of atmospheric nitrogen deposition are difficult to make because of uncertainties in estimating deposition, especially dry deposition, as well as watershed nitrogen retention.[24] Paerl (1993) reviews the importance of atmospheric nitrogen deposition as a contributor to eutrophication of coastal ecosystems; he concludes that 10 to 50 percent of the total nitrogen loading to coastal waters is from direct and indirect atmospheric deposition. Estimates for the economically important Chesapeake Bay indicate that about 25 to 40 percent of the nitrogen loadings to the bay occur via atmospheric deposition.[25] Hinga et al. (1991) estimate that anthropogenic deposition provides 11 percent of total anthropogenic inputs of nitrogen in Narragansett Bay, 33 percent for the New York Bight, and 10 percent for New York Bay. Fisher and Oppenheimer (1991) estimate that atmospheric nitrogen provides 23 percent of total nitrogen loading to Long Island Sound and 23 percent to the lower Neuse River in North Carolina. Information on the importance of atmospheric nitrogen deposition for most other U.S. coastal ecosystems is not available in the literature. Episodic atmospheric inputs of nitrogen may be an important source of nitrogen to nutrient-poor marine ecosystems, such as the North Atlantic near Bermuda and the North Sea.[26]

Valuing Potential Benefits from Eutrophication Avoidance Under the CAA

It is currently estimated that in the absence of pollution reductions achieved under the Clean Air Act, total nitrogen emissions to the atmosphere would have increased by nearly 90 million tons by 1990, a two-fold increase above 1990 levels estimated with CAA controls remaining in place.[27] However, the ability to determine the potential economic benefit from such a reduction in nitrogen emissions is heavily constrained by gaps in our current biological and economic knowledge base of aquatic ecosystems.

One water body that has received much study in the area of nitrogen-induced eutrophication is Chesapeake Bay. As previously discussed, it is estimated that atmospheric deposition of nitrogen contributes approximately 25 percent to the total nitrogen load-

[19] Paerl, 1993.

[20] Paerl, 1988.

[21] Hansson and Rudstam, 1990; Rosenberg et al., 1990; Paerl, 1993.

[22] Hecky and Kilham, 1988; Vitousek and Howarth, 1991.

[23] U.S. EPA, 1993; Paerl, 1993.

[24] U.S. EPA, 1993.

[25] U.S. EPA, 1994.

[26] Owens et al., 1992.

[27] U.S. EPA, 1995; Table B-3.

ings to the bay.[28] In deposition terms, an estimated 15 to more than 25 percent increase in total nitrogen deposition has been forecast in the Chesapeake Bay watershed by 1990 in the absence of CAA pollution controls.[29] These results are based on an estimated 40,000 tons of atmospherically deposited nitrogen (as nitrate and ammonia) to Chesapeake Bay in 1985,[30] which means a 20 percent increase in atmospheric deposition would amount to approximately 8,000 additional tons.

One indirect method available to gauge the potential economic relevance of avoidance of such atmospheric nitrogen loadings to Chesapeake Bay is through the avoidance cost of nitrogen controls. However, such an assessment is difficult due to the site, facility, and treatment-specific variation in treatment costs. For example, Camacho (1993) reviewed nitrogen treatment costs for chemical treatment of water from important point sources (mostly public owned treatment works) and found that costs ranged from \$9,600 to \$20,600 per ton (annual costs, 1990 dollars), depending on the facility evaluated. Biological treatment of nitrogen from point sources was far more expensive, varying from \$4,000 to \$36,000 per ton. For control of non-point source loading, values of nitrogen removal practices ranged from \$1,000 to \$285,000 per ton.[31] Taking chemical addition as one possible example, the avoided costs of treatment of 8,000 tons of nitrogen would range from about \$75 million to about \$170 million annually (in 1990 dollars).

Mercury

Mercury, in the form of methyl mercury, is a neurotoxin of concern and can accumulate in tissue of fish to levels that are hazardous to humans and aquatic-feeding wildlife in the U.S. In relation to the section 812 CAA retrospective analysis, mercury is of interest for two reasons. First, potential benefits to human welfare may have occurred as a result of mercury emission controls implemented under EPA's National Emission Standards for Hazardous Air Pollutants (NESHAP). Second, experimental and observational evidence suggests that acidification of water bodies enhances mercury accumulation in fish tissues.[32] Therefore, CAA-mandated reductions in sulfur and nitrogen oxide emissions and subsequent acid deposition may have resulted in indirect benefits from a reduction in mercury accumulation in fish and subsequent improvements to human health and welfare.

The accumulation of mercury to hazardous levels in fish has become a pervasive problem in the U.S. and Canada. A rapid increase in advisories occurred during the 1980s, including a blanket advisory affecting 11,000 lakes in Michigan.[33] The Ontario Ministries of Environment and Natural Resources (1990) recommend fish consumption restrictions for 90 percent of the walleye populations, 80 percent of smallmouth bass populations, and 60 percent of lake trout populations in 1,218 Ontario lakes because of mercury accumulation. In many instances, mercury has accumulated to hazardous levels in fish in highly remote water bodies that are free from direct aqueous discharges of mercury.[34] Mass balance studies have shown that atmospheric deposition of mercury can account for the accumulation of mercury in fish to high levels in lakes of these remote regions.[35] The potential impacts of mercury on the health of humans and fish-eating (piscivorous) wildlife has lead EPA to recently establish water quality criteria to protect piscivorous species in the Great Lakes.[36]

Although mercury accumulation in fish via atmospheric deposition is now widely recognized as a potential hazard to human health and certain wildlife species, studies establishing quantitative linkages between sources of mercury emissions, atmospheric deposition of mercury, and subsequent accumulation in fish are lacking. Thus at the present time, we are unable to quantify potential benefits from CAA-avoided mercury accumulation in fish of U.S. water

[28] U.S. EPA, 1993.

[29] U.S. EPA 1995, Figure C-6.

[30] NERA, 1994.

[31] Shuyler, 1992.

[32] Bloom et al., 1991; Watras and Bloom, 1992; Miskimmin et al., 1992; Spry and Wiener, 1991; Wiener et al., 1990.

[33] Watras et al., 1994.

[34] Glass et al., 1990; Sorenson et al., 1990; Grieb et al. 1990; Schofield et al. 1994.

[35] Fitzgerald et al. ,1991.

[36] U.S. EPA, 1995.

bodies. Given the pervasiveness of the mercury problem with CAA-pollution controls, potential benefits to human health and welfare from avoidance of further mercury related damages to aquatic ecosystems could be substantial.

It should also be noted that atmospheric deposition is a major contributor to surface water loads of other toxic pollutants as well. For example, scientists believe that about 35 to 50 percent of the annual loadings of a variety of toxic chemicals to the Great Lakes may be from the air; for lead, atmospheric deposition currently accounts for an estimated 95 percent of the total load in the Great Lakes.[37] CAA-related reductions in air emissions of toxic pollutants (such as lead) undoubtedly reduced the loading of these chemicals to the Great Lakes and other water bodies; the magnitude of the benefits of reducing these exposures to humans and wildlife is not known.

Benefits from Avoided Damages to Wetland Ecosystems

Introduction

This review addresses the effects of air pollutants on wetland ecosystems; the focus is on acidification and nutrient loading. Valuable service flows of wetland ecosystems include flood control, water quality protection and improvement, wildlife and fish habitat, and biodiversity. The limited scientific evidence suggests that air pollutants may most affect biodiversity, in particular because of nutrient loading through nitrogen deposition.

Wetlands are broadly characterized as transitional areas between terrestrial and aquatic systems in which the water table is at or near the surface or the land is periodically covered by shallow water.[38] Types of wetlands include swamps (forested wetlands), marshes (herbaceous vegetation), and peatlands, which are wetlands that accumulate partially decayed vegetative matter due to limited decomposition.[39] Peatlands include bogs and fens. Bogs receive water solely from precipitation, are generally dominated by *Sphagnum* moss, and are low in nutrients. Fens receive water from groundwater and precipitation, contain more marsh-like vegetation, and have higher pH and nutrient levels than bogs.[40] Most of the limited work on the effects of atmospheric deposition on wetlands has been done in peatlands, specifically in Europe, where levels of atmospheric deposition are generally much higher than in the U.S.

The air pollutants of greatest concern with respect to effects on wetland ecosystems are oxides of nitrogen (NO_x) and oxides of sulfur (SO_x), primarily sulfur dioxide (SO_2). Air pollutants may affect wetland ecosystems by acidification of vulnerable wetlands and by increasing nutrient levels. Acidification in vulnerable wetlands may affect vegetation adversely, as appears to have occurred in Europe. In wetlands where nitrogen levels are low, increased nitrogen deposition may alter the dynamics of competition between plant species. Species adapted to low-nitrogen levels, including many endangered species, may decrease in abundance.[41]

Effects of Acidification

Limited evidence suggests that acidic deposition and decreased pH may harm certain wetland plants, alter competitive relations between wetland plants and cause changes in wetland drainage and water retention.

Work concerning the possible acidification of peatlands is inconclusive. Acidic deposition is unlikely to result in displacement of base cations from cation exchange sites in bogs, and therefore it will not cause a drop in pH.[42] Peatland sediments are low in Al^{3+}, so mobilization of toxic aluminum is not a concern as it is in forest soils and aquatic ecosystems.[43] Acidification might affect certain fen ecosystems. Gorham et al. (1984) have hypothesized that acidic deposition could leach base cations from mineral-poor fens and decrease pH levels. This could result in a

[37] U.S. EPA, 1994.

[38] Cowardin et al., 1979.

[39] Mitsch and Gosselink, 1986.

[40] Mitsch and Gosselink, 1986.

[41] U.S. EPA, 1993.

[42] Gorham et al., 1984.

[43] Turner et al., NAPAP SOS/T 10, 1990.

transition to bog vegetation such as *Sphagnum* and away from sedge meadow vegetation. At this time, this remains a hypothesis; however, pH did not decrease in a mineral-poor Ontario fen during a four-year period in which researchers experimentally increased acidic deposition.[44]

In European wetlands affected by high levels of deposition for many years, acidic deposition has seriously affected wetland vegetation. Roelofs (1986) reports that acidification of heath pools in the Netherlands has caused a change in species composition with *Sphagnum* and rushes replacing the original vegetation. Likewise, significant declines in *Sphagnum* in British bogs have occurred in areas affected by 200 years of atmospheric pollution, including nitrogen deposition.[45] It is unclear how such changes have affected wetland service flows apart from the effects on biodiversity; however, water retention has decreased and significant erosion has occurred in seriously perturbed British bogs near Manchester and Liverpool.[46]

Effects of Nutrient Loading

Atmospheric deposition may affect wetlands by increasing the level of nutrients, particularly nitrogen, in wetlands. Sulfur is not a limiting nutrient in peatlands,[47] but nitrogen commonly limits plant growth.[48] The effects of increased nitrogen levels in wetlands include an increased threat to endangered plant species and possible large-scale changes in plant populations and community structure. Endangered and threatened plant species are common in wetlands, with wetland species representing 17 percent of the endangered plant species in the U.S. (U.S. EPA, 1993). These plants are often specifically adapted to low nitrogen levels; examples include isoetids[49] and insectivorous plants.[50] In eastern Canadian wetlands, nationally rare species are most common in infertile sites.[51] When nitrogen levels increase, other species adapted to higher levels of nitrogen may competitively displace these species. Thus, NO_x emissions that increase nitrogen levels in nitrogen-poor wetlands may increase the danger of extinction for threatened and endangered species.

By changing competitive relations between plant species, increased nitrogen deposition may broadly affect community structure in certain wetlands. Common species that thrive in nitrogen-poor wetlands may become less abundant. Many nitrogen-poor bogs in the northern U.S. are dominated by *Sphagnum* species. These species capture low levels of nitrogen from precipitation. Increased nitrogen levels may directly harm *Sphagnum* and cause increased nitrogen to be available to vascular plants that may out compete *Sphagnum.*[52] Studies in Great Britain have documented large declines in *Sphagnum* moss because of atmospheric pollution;[53] nitrogen loading may play an important role in these declines. However, Rochefort et al. (1990) document limited effects of fertilization from experimentally-increased NO_3^- and SO_4^{2-} deposition on an Ontario mineral-poor fen over a four-year period, apart from initially increased *Sphagnum* growth. Thus, increased nitrogen loading might adversely or beneficially affect wetland plants depending on baseline nitrogen concentrations in the wetland, atmospheric nitrogen loading, and species requirements for and sensitivity to nitrogen.

Increases in nitrogen levels due to NO_x emissions will have the greatest effect on wetlands that are extremely nitrogen-limited and that receive small amounts of nitrogen naturally. Since bogs, including *Sphagnum* bogs, receive little surface water runoff, they get most of their nutrient and water loadings through precipitation. These bogs may receive a total of approximately 10 kg nitrogen per hectare per year (kg N/ha/yr), which is one to two orders of magnitude less nitrogen than other freshwater wetlands and

[44] Rochefort et al., 1990.

[45] Lee et al., 1986.

[46] Lee et al., 1986.

[47] Turner et al., NAPAP SOS/T 10, 1990.

[48] U.S. EPA, 1993.

[49] Boston, 1986.

[50] Moore et al., 1989.

[51] Moore et al., 1989; Wisheu and Keddy, 1989.

[52] Lee & Woodin 1988, Aerts et al., 1992.

[53] Ferguson et al., 1984; Lee et al., 1986.

saltmarshes receive.[54] As atmospheric deposition of nitrogen has been estimated to be at least 5.5 to 11.7 kg N/ha/yr,[55] changes in NO_x emissions would most likely affect these bogs. The results of a model by Logofet and Alexandrov (1984) suggest that a treeless, nutrient-poor bog may undergo succession to a forested bog because of the input of greater than 7 kg N/ha/yr.

As in freshwater wetlands, significantly increased nitrogen deposition to coastal wetlands will increase productivity and alter the competitive relationships between species.[56] However, studies showing this increased productivity have used 100 to 3000 kg N/ha/yr.[57] Therefore, limited changes in NO_x emissions may not affect coastal wetland productivity.

Summary of Wetland Ecosystem Effects

The effects of air pollutants on wetlands have received little attention, in contrast to the large body of work on the effects of acid rain on aquatic and forest ecosystems. Little evidence exists suggesting that acidification due to atmospheric deposition is a major threat to wetlands. In particular, peatlands are naturally acidic, although mineral-poor fens may be at risk from acidification. Nitrogen loading may alter community composition in wetlands naturally low in nutrients, such as bogs. Nitrogen loading may threaten rare species adapted to low nitrogen levels. In Britain and The Netherlands, heavy atmospheric deposition over a long period appears to have caused serious declines in *Sphagnum* in peatlands.

Air pollutants appear to most seriously threaten rare and endangered species, biodiversity, and community composition in wetlands, particularly bogs. These changes are difficult to associate with changes in economic value; even the qualitative nature of the effects is uncertain. Air pollutants may not significantly affect such important wetland service flows as flood control, water quality protection, and wildlife habitat in most wetlands, so the impacts on the more readily monetized aspects of the economic value of wetlands may be limited.

Benefits from Avoided Damages to Forests

Introduction

Forests occupy 33 percent of the land mass in the U.S. (some 738 million acres) and provide a wealth of services to the U.S. population.[58] Notable services provided by forests include timber production, recreational opportunities such as hunting, camping, hiking, and wildlife observation, water quality protection, nutrient removal and cycling, flood control, erosion control, temporary carbon sequestration, preservation of diversity, and existence values. In 1991, hunting participation alone accounted for 236 million recreation days that included 214 million person trips with estimated expenditures valued at $12.3 billion.[59]

The Clean Air Act-regulated pollutants of greatest concern with respect to effects on forest ecosystems are oxides of sulfur (SO_x), primarily sulfur dioxide (SO_2), oxides of nitrogen (NO_x), and volatile organic compounds (VOCs). While extremely high ambient concentrations of SO_2 and NO_x may directly affect vegetation, such effects are uncommon in the U.S.;[60] the indirect effects of these pollutants are of greater concern. Specifically, emissions of SO_2 and NO_x are known to contribute to acid deposition in portions of the United States, with SO_2 contributing 75 to 95 percent of the acidity in rainfall in the eastern U.S.[61] Acid deposition is of concern to forests primarily from the acidification of soils (i.e., by reducing seed germination, altering nutrient and heavy metal availability). Direct foliar damage can occur from precipitation with extremely low pH levels (i.e., 3.0-3.6 and below), although these levels are lower than ambient levels in the U.S.[62] VOCs and NO_x are

54 U.S. EPA, 1993.

55 U.S. EPA, 1993.

56 U.S. EPA, 1993.

57 U.S. EPA, 1993.

58 Powell et al. 1993.

59 U.S. DOI, 1993.

60 Shriner et al., NAPAP SOS/T 18, 1990.

61 NAPAP, 1991.

62 Shriner et al., NAPAP SOS/T 18, 1990.

important precursors to ozone formation, which can affect leaf photosynthesis and senescence and decrease cold hardiness, thereby causing deleterious impacts on tree growth, survival and reproduction. Deposition of NO_x may also alter the nutrient balance of forest soils, which in turn might alter the competitive relationships between tree species and affect species composition and diversity.[63]

Current Air Pollutant Effects on Forests

Acid Deposition Impacts

In 1985, NAPAP organized the Forest Response Program (FRP) to evaluate the significance of forest damage caused by acidic deposition, the causal relationships between air pollutants and forest damage, and the dynamics of these relationships regionally. Research was focussed on four forest regions: Eastern Spruce-Fir, Southern Commercial Forests, Eastern Hardwoods, and Western Conifers. With the exception of high-elevation spruce-fir forests, the available evidence suggests that acidic deposition does not currently affect these forests and that observed declines in sugar maple and southern pines are not due to acidic deposition.[64]

Circumstantial evidence suggests that acidic deposition may affect high-elevation spruce-fir forests in the northeastern U.S. These forests have extensive contact with acidic cloud water.[65] Experimental evidence suggests that acidic deposition may affect cold hardiness in red spruce, an important component of the spruce-fir forest. Significant declines in red spruce growth and in its importance in the forest have occurred in New York and northern New England. The proximate cause of death of red spruce in the region is pathogens and insects; acidic deposition may interact with these biological stresses and with weather-induced stress to produce adverse effects in red spruce. Ozone may also play a role in red spruce decline in this region.[66] Available evidence suggests that soil aluminum and soil pH levels have not affected red spruce adversely.[67]

Ozone Impacts

Experimental Evidence

For practical reasons, the majority of experimental evidence linking ozone exposure to damage to tree species has been derived from studies of individual plants, especially seedling and branch studies.[68] Results from these studies suggest that ozone exposure can reduce photosynthesis and increase senescence in leaves. Subsequently, such effects from ozone may alter the carbohydrate allocation to plant tissues such as roots, which may affect plant growth and cold hardiness. Decreases in cold tolerance may be particularly important for trees in northern latitudes and high elevations. Recent work on quantifying the relationship between ozone exposure and plant responses suggest that seedlings of aspen, ponderosa pine, black cherry, tulip poplar, sugar maple, and eastern white pine seedlings may experience biomass reductions of approximately 10 percent at or near ambient ozone exposures.[69] Because trees are perennials, the effect of even a 1-2 percent per year loss in seedling biomass (versus 10 to 20 percent yield loss in crops), if compounded over multiple years under natural field conditions of competition for resources, could be severe.

Although indicative of short-term relative response to ozone exposure, results from these experiments are unable to provide reliable information on the long-term effects of ozone on forests. This limitation arises because the effects of ozone on forests will depend on both the response of individual plants to ozone exposure and the response of populations of plants, which interact with their environment. Population response will be altered by the varying intraspecific genetic susceptibility to ozone. Individual plant response will also be affected by many environmental factors, including insect pests, pathogens, plant symbionts, competing plants, moisture, temperature, light, and other pollutants. Consistent evidence on the interaction of ozone with other environmental factors is lacking. Furthermore, most experimental stud-

[63] U.S. EPA, 1993.

[64] Barnard et al., NAPAP SOS/T 16, 1990; NAPAP, 1991.

[65] Barnard et al., NAPAP SOS/T 16, 1990.

[66] Shriner et al., NAPAP SOS/T 18, 1990.

[67] Barnard et al., NAPAP SOS/T 16, 1990.

[68] U.S. EPA, 1996a.

[69] Hogsett et al. , 1995.

ies have only studied exposure for one growing season; effects on forest species may occur over decades.[70] Therefore, considerable uncertainties occur in scaling across individuals of different ages, from individuals to populations and communities, and across time.

Observational Evidence

Studies of the forests of the San Bernardino Mountains provide the strongest case for linking ozone exposure to damages to an entire forest ecosystem. These forests have been exposed to extremely high ambient ozone levels over the past 50 years due to their proximity to the Los Angeles area. The area has been extensively studied regarding the effects of ozone, as described in U.S. EPA (1996a). The ecosystem has been seriously affected by ozone pollution, with the climax-dominant, but ozone-sensitive ponderosa pine and Jeffrey pine declining in abundance, replaced by more ozone-tolerant species. These sensitive species have experienced decreased growth, survival, and reproduction, and susceptibility to insects. The effects of ozone on these species have resulted in other ecosystem effects, including the buildup of a large litter layer, due to increased needle senescence. The decline of the fire-tolerant ponderosa and Jeffrey pines may seriously affect the fire ecology of the ecosystem, with fire-sensitive species becoming more common. Ozone concentrations have been declining in recent decades, and crown injury of ponderosa and Jeffrey pine has decreased. However, the two species have continued to decline in abundance, as measured by total basal area, compared with other species over the period 1974 to 1988.[71] The nature of community dynamics, particularly in mixed species, uneven aged stands, indicates that subtle long-term forest responses (e.g., shifts in species composition) to elevated levels of a chronic stress like exposure to ozone are more likely than wide-spread community degradation.[72]

Limited field studies have been completed in other forest ecosystems. Foliar injury has been observed in the Jefferson and George Washington National Forests and throughout the Blue Ridge Mountains, including areas of the Shenandoah National Park.[73] In the Great Smoky Mountains National Park, surveys made in the summers from 1987 through 1990 found 95 plant species exhibited foliar injury symptoms consistent with those thought to be caused by ozone.[74] Foliar ozone injury has also been documented in National Parks and Forests in the Sierra Nevada mountains.[75]

Growth and productivity of seedlings have been reported to be affected by ozone for numerous species in the Blue Ridge Mountains of Virginia. In the Shenandoah National Park, Duchelle et al. (1982, 1983) found that tulip poplar, green ash, sweet gum, black locust, as well as several evergreen species (e.g., Eastern hemlock, Table Mountain pine, pitch pine, and Virginia pine), common milkweed, and common blackberry all demonstrated growth suppression of seedlings. Except for the last two species mentioned, almost no visible injury symptoms accompanied the growth reductions. Studies of mature trees in the Appalachian Mountains also indicate that injury associated with exposure to ozone and other oxidants has been occurring for many years.[76] Researchers have also found that major decreases in growth occurred for both symptomatic and asymptomatic trees during the 1950s and 1960s in the Western U.S.[77] The adverse response of a number of fruit and nut trees to ozone exposure has been reported.[78]

Monitoring by the USDA Forest Service shows that growth rates of yellow pine in the Southeast have been decreasing over the past two decades in natural stands but not in pine plantations.[79] Solid evidence linking this growth reduction to air pollutants is lack-

[70] U.S. EPA, 1996a.

[71] Miller et al., 1989 and Miller et al., 1991.

[72] Shaver et al., 1994

[73] Hayes and Skelly, 1977; Skelly et al., 1984

[74] Neufeld, et al., 1992

[75] Peterson and Arbaugh, 1992

[76] Benoit et al., 1982

[77] Peterson et al., 1987; Peterson and Arbaugh, 1988, 1992; Peterson et al., 1991

[78] McCool and Musselman, 1990; Retzlaff et al., 1991, 1992a, b

[79] NAPAP, 1991.

ing, although ozone, in particular, may be a factor.[80] Ambient ozone levels in the region are high enough to damage sensitive tree species, including pine seedlings during experimental exposure.[81] Due to the commercial importance of yellow pine, the economic impacts of ozone on forest ecosystems in this area could be significant if ozone is affecting growth.

Although the ecosystem effects occurring in the San Bernardino forest ecosystem have occurred at very high ozone exposures, lower ozone exposure elsewhere in the U.S. may still affect forests. The EPA Ozone Staff Paper[82] assessed the risk to vegetation, including forests, under current ambient air quality. Using a GIS approach, it was found that under the base year (1990) air quality, a large portion of California and a few localized areas in North Carolina and Georgia have seasonal ozone levels above those which have been reported to produce greater than 17 percent biomass loss in 50 percent of studied tree seedling species. A broader multistate region in the east is estimated to have air quality sufficient to cause 17 percent biomass loss in seedlings, while at least a third of the country, again mostly in the eastern U.S., most likely has seasonal exposure levels which could allow up to 10 percent yield loss in 50 percent of studied seedlings. The Staff Paper did not present monetized benefits because of lack of exposure-response functions.[83]

Even small changes in the health of ozone-sensitive species may affect competition between sensitive and tolerant species, changing forest stand dynamics.[84] Depending on the sensitivities of individual competing species, this could affect timber production either positively or negatively, and affect community composition and, possibly, ecosystem processes.

Endangered species

Ozone effects may also reduce the ability of affected areas to provide habitats to endangered species. For example, two listed endangered plant species, the spreading aven and Roan Mountain bluet, are currently found at a small number of sites in eastern Tennessee and western North Carolina — forested areas where ozone-related injury is of concern.[85] In addition, ozone-related effects on individual ozone-sensitive species that provide unique support to other species can have broader impacts. For example, one such species is the common milkweed, long known for its sensitivity to ozone and usefulness as an indicator species of elevated ozone levels, as well as being the sole food of the monarch butterfly larvae. Thus, a major risk associated with of the loss of milkweed foliage for a season is that it might have significant indirect effects on the monarch butterfly population. A large number of studies have shown that ozone-sensitive vegetation exists over much of the U.S., with many native species located in forests and Class I areas, which are federally mandated to preserve certain air quality related values.

Valuation of Benefits From CAA-Avoided Damages to Forests

Background

To quantitatively assess the economic benefits of avoided damages of relevant CAA pollutants to forests, it is necessary to link estimated changes in air pollution to measures of forest health and conditions that can be readily quantified in economic terms. For commercial timber production, this would require quantifying the relationship between atmospheric deposition and measures of forest productivity such as timber yield. For assessing recreational benefits, linkages would have to be drawn between air pollution and vulnerable factors that influence forest-based recreation (e.g., site-characteristics such as canopy density, type of tree species, degree of visible tree damage, etc.). While important strides have been made in establishing these linkages (e.g., NAPAP modeling of air pollution effects on forest soil chemistry and tree branch physiology), critical gaps in our ability to predict whole tree and forest responses to air pollution changes have precluded the establishment of such quantitative linkages.[86] Critical knowl-

[80] NAPAP, 1991.

[81] NAPAP, 1991.

[82] U.S. EPA, 1996b

[83] U.S. EPA, 1996b.

[84] U.S. EPA, 1996a.

[85] U.S. EPA, 1996b

[86] NAPAP, 1991.

edge gaps exist in our ability to extrapolate experimental results from seedling and branch studies to whole tree and forest responses, to account for key growth processes of mature trees, to integrate various mechanisms by which air pollution can affect trees (e.g., soil acidification, nitrification, and direct foliar damage, winter stress, etc.), and to account for the interaction of other stressors on forest health and dynamics (susceptibility to insect damage, drought, disease, fire, nutrient and light competition, etc.).

Despite these constraints to quantifying economic benefits from air pollution reductions on forest ecosystems, relevant studies that have attempted to value air pollution damages on forests are reviewed and summarized below. In some cases, the relationship between air pollution and forest response is estimated using expert judgement (e.g., for NAPAP assessment from various growth scenarios). In other cases, damage estimates reflect current impacts of air pollution on forests, and the dose-response relationship is absent. In the aggregate, this summary provides some insight into possible CAA-related benefits from avoided damages to a select and narrowly focussed group of forest services, but, because of severe data constraints, does not provide an estimate of the overall range of forest-based benefits possible under the CAA.

Commercial Timber Harvesting

The economic impact of hypothetical growth reductions in northeastern and southeastern trees (both hardwood and softwood species) was intensively studied under NAPAP.[87] Growth reductions ranging from 5 to 10 percent over a 5 to 10 year period, depending on the species and location, were assumed to occur as a result of all forms of air pollution based on expert opinion derived from a survey by deSteigner and Pye (1988). Timber market responses to these hypothesized growth declines were modeled until the year 2040 using a revised version of the Timber Assessment Market Model (TAMM90) and the Aggregate Timberland Assessment System (ATLAS), which was used to simulate timber inventories on private timberland in the United States. Economic welfare outputs included changes in consumer and producer surplus and changes in revenue to southeast stumpage owners. Results indicate that annualized reductions in consumer and producer surplus would total $0.5 billion by the year 2000 and $3 billion by the year 2040 (in 1967 dollars). Simulated effects on stumpage owners' revenues were minimal ($10 to $20 million).

In an attempt to estimate the net economic damages from ozone effects on selected U.S. forests, NAPAP studied the effect of various *assumed* reductions in growth rates of commercial southeastern pine forests (both natural and planted).[88] For both planted and natural plus planted pines, the following changes in growth rates were assumed to occur: a two percent increase, no change, a two percent decrease, a five percent decrease, and a ten percent decrease. The two to five percent growth reductions were considered as possible outcomes from current ozone induced damage to southeastern forests, although no quantitative linkage between ozone exposure and damages was established. The ten percent growth reduction scenario was primarily included for evaluating model sensitivity to severe changes in growth and was considered out of the range of likely ozone damage estimates. The TAMM and ATLAS models were again used to simulate timber market responses under baseline and hypothesized growth change scenarios from 1985 to 2040. Results indicate that *annual* changes in total economic surplus (i.e., the sum of consumer and producer surplus and timber owner revenues in 1989 dollars) would range from an increase of $40 million (for the two percent increase in growth scenario) to a decrease of $110 million (for the ten percent decrease in growth scenario) for planted and natural pine model simulations.

In the context of estimated benefits from avoidance of other damages in the absence of the Clean Air Act from 1970 to 1990,[89] the magnitude of economic damages estimated to the commercial timber industry are comparatively small. For example, economic damage estimates range up to $3 billion annually for five to ten percent growth rate reductions in northeast and southeast forests, and just $110 million for southeastern pines. However, in the context of damages to forest-based services as a whole, the NAPAP-derived commercial timber damage estimates should be viewed as representing a lower bound estimate for a variety of reasons. First, these damage estimates exclude other categories of possible forest-based ben-

[87] Haynes and Kaiser, NAPAP SOS/T 27 Section B, 1990.

[88] NAPAP, 1991.

[89] Most notably avoided human health effects, which are estimated on the order of $300 to $800 billion annually.

efits, including recreational and non-use values. Second, even within the context of timber-related damages, the NAPAP forest-damage studies focused on a portion of U.S. forests (northeastern and southeastern U.S.); a much greater geographic range of forests could become susceptible to timber-related damages in the absence of CAA controls. Finally, the NAPAP damage estimates consider only two types of tree species: planted and naturally grown pines, although these species are economically important. Damages to other commercially harvested tree species, such as mixed pine and hardwood forests, are therefore excluded.

Non-marketed Forest Services

In an effort to address the potential benefits resulting from avoidance of acid deposition-induced damages to non-marketed forest-based services (e.g., recreation use, existence value), an extensive review of the economic literature was conducted under the auspices of NAPAP.[90] From their review, NAPAP could not identify any single study or model that could be reliably used to quantify economic benefits from avoided acid deposition-caused damages to non-marketed forest services (such as recreational use) on a regional or national basis. The primary limitation in many of the studies reviewed was the absence of a quantitative linkage between the value of a recreational user day and important site characteristics which could be tied to air pollution effects. In addition, most studies were narrowly focused geographically to specific sites and did not attempt to value system-wide (larger scale) damages that could result from acid deposition over an entire region. Since the availability of nearby substitution sites will affect the recreational value for a given site, the benefits from such site-specific studies may not reflect actual economic damages incurred from wide-scale air pollution impacts on forests. The inability of studies to consider additional crowding at unaffected sites in addition to changes in recreational participation rates as a function of air pollution damages was also recognized as an important limitation.

Despite not being able to quantitatively assess the benefits from avoided acid deposition-induced damages to nonmarket forest services, several important concepts emerge from NAPAP's review of recreational benefits, that bear relevance to the section 812 retrospective analysis. First, several studies were identified that established a relationship between key forest site characteristics and the value of recreational participation. For example, Brown et al. (1989) used contingent valuation to evaluate the relationship between scenic beauty ratings and willingness of recreationalists to pay at pictured sites. Based on their interviews with over 1400 recreationalists at ten different sites in Arizona, positive correlations were established between scenic beauty rankings determined from one group of recreationalists and willingness to pay to recreate determined by a separate group of recreationalists (r^2 ranged from 0.27 to 0.98 depending on ranking). In another study, Walsh et al. (1989) developed a functional relationship between reduction of recreational benefits and tree density changes that reflected varying levels of insect damage at six campgrounds in the Front Range of the Colorado Rockies. By using both contingent valuation and travel cost models, Walsh et al. (1989) were able to show that 10 percent, 20 percent, and 30 percent decreases in tree densities reduces the total recreational benefits at their sites by 7 percent, 15 percent and 24 percent, respectively. Although results from these studies are limited to the sites from which they were derived, they do support the intuition that the degree of visible damage to forests is to some extent correlated with the magnitude of damages to forest-based recreation expected. This finding supports the notion that the avoidance of damages to forest ecosystems from CAA-induced pollution controls (albeit currently unquantified) have likely benefited forest-based recreation in the U.S.

In addition to establishing relationships between recreational value and visible damage to forest sites, there is evidence linking air pollution (ozone) effects on forests to economic damages to non-use values of forests. For example, D.C. Peterson et al. (1987) valued ozone-induced damages to forests surrounding the Los Angeles area. Using contingent valuation methods, D.C. Peterson et al. (1987) surveyed recreationalists (a random survey of households in the San Bernardino, Los Angles and Orange counties) and residents (a sample of property owners within the San Bernardino and Angeles national forests) for their willingness to pay to prevent forest scenes from degrading one step on a "forest quality ladder" depicting various levels of ozone-induced damages. The mean willingness to pay to protect further degradation was $37.61 and $119.48 per household for recreationalists and residents, respectively. Annual damages to Los Angeles area residences from a one-step drop on the forest quality ladder were estimated between $27 million and $147 million.

[90] Rosenthal, NAPAP SOS/T 27 Section B, 1990.

These estimates cannot be directly translated into a rough estimate of the potential non-use values of avoided forest damages. Considering the limited size of the population generating the estimated benefits of forest degradation, however, they do provide evidence that the recreational and non-use benefits may substantially exceed the commercial timber values.

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